The carbon footprint of pasture-based milk production: Can white clover make a difference?

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ABSTRACT

Carbon footprint (CF) calculated by life cycle assessment (LCA) was used to compare greenhouse gas emissions from pasture-based milk production relying mainly on (1) fertilizer N (FN), or (2) white clover (WC). Data were sourced from studies conducted at Solohead Research Farm in Ireland between 2001 and 2006. Ten FN pastures stocked between 2.0 and 2.5 livestock units (LU)/ha with fertilizer N input between 180 and 353 kg/ha were compared with 6 WC pastures stocked between 1.75 and 2.2 LU/ha with fertilizer N input between 80 and 99 kg/ha. The WC-based system had 11 to 23% lower CF compared with FN (average CF was 0.86 to 0.87 and 0.97 to 1.13 kg of CO₂-eq/kg of energy-corrected milk, respectively, 91% economic allocation). Emissions of both N₂O and CO₂ were lower in WC, whereas emissions of CH₄ (per kg of energy-corrected milk) were similar in both systems. Ratio sensitivity analysis indicated that the difference was not caused by error due to modeling assumptions. Replacing fertilizer N by biological nitrogen fixation could lower the CF of pasture-based milk production.

Key words: carbon footprint, life cycle assessment, white clover, milk production

INTRODUCTION

Because of projected population growth and demand for dairy products (Steinfeld et al., 2006), urgent action is needed to achieve a sustainable balance between profitability and the environmental impact of dairy production. The global dairy sector was estimated to contribute 4% of anthropogenic greenhouse gas (GHG) emissions in 2007 (Gerber et al., 2010). In countries with small human populations and large cattle populations such as Ireland, agriculture is the largest contributor (30.5% in Ireland) to GHG emissions (Ireland EPA, 2012). As milk production accounts for over one-third

of the output of Irish agricultural commodities (Anonymous, 2011), GHG emissions from milk are important to policy makers. Tools are needed to assist with strategic policy development to enable the dairy sector to thrive while minimizing GHG emissions.

Life cycle assessment (LCA; ISO, 2006) has been developed to assess the environmental impact through the life cycle of products, from the “cradle” (production of raw materials such as iron ore) to the “grave” (the waste management of products after consumption). When applied to agricultural products, attention is often focused on “cradle to farm gate” because the greatest impact is found in the production stage (Schau and Fet, 2007). Because of global concerns about GHG emissions from livestock production, the LCA interpretation of GHG emissions is performed more often than other impact categories (e.g., eutrophication) and is referred to as carbon footprint (CF; O’Brien et al., 2010; Rotz et al., 2010; Flysjö et al., 2011). The main GHG from agriculture are carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O). For pasture-based milk production, mineral fertilizer and recycled organic manures are the main N inputs to grassland and the main sources of N₂O emissions from farms. Typical management in grazing systems uses mineral fertilizer N (FN) as the predominant source of N for grassland (referred to hereafter as FN management) in addition to manure.

In temperate pastures, biological N fixation (BNF) from forage legumes can also be a significant source of N (10 to 300 kg of N/ha per year; Ledgard et al., 2009). Because of increasing fertilizer prices and stringent regulation of N use on farms (European Council, 1991), white clover (*Trifolium repens* L.) is becoming an increasingly profitable alternative to FN for pasture-based dairy production (Humphreys et al., 2012). Management of white clover (WC) in grassland (hereafter referred to as WC management) provides BNF, which displaces the need for fertilizer N and the GHG emissions associated with fertilizer N in the system. Measurement under field conditions at Solohead Research Farm (Co. Tipperary, Ireland) showed that use of WC reduced N₂O emissions by 19% (Li et al., 2011). Stud-

Received July 4, 2012.

Accepted October 19, 2012.

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Table 1. Characteristics of the milk production based on fertilizer N (FN)- and white clover (WC)-based systems at Solohead Research Farm (Co. Tipperary, Ireland) between 2001 and 2006 (Humphreys et al., 2008, 2009)¹

Characteristic	2001–2002 ²				2003–2006	
	WC	FN				
	N205	N230	N300	N400	WC	FN
Stocking rate, ³ LU/ha	1.75	2.1	2.5	2.5	2.0; 2.2	2.0; 2.2
Synthetic fertilizer N, kg/ha	80	180	248	353	90	226
Concentrate feed, kg/cow per year	536	536	536	536	520	531
Milk delivered at farm gate, kg/cow	6,550	6,275	6,242	6,375	6,521	6,526
Milk fat, %	4.1	4.2	4.1	4.2	4.2	4.2
Milk protein, %	3.5	3.6	3.5	3.6	3.5	3.6
Biological-N fixation, kg/ha per year	87.4	9.1	3.2	0.1	112.5	12.2

¹Data are means of 2 and 4 yr, respectively.

²The acronyms for 2001 and 2002 are consistent with Humphreys et al. (2008); no acronyms were previously defined for the experiments between 2003 and 2006.

³The stocking rate (LU = livestock unit) was 2 for 2003 and 2.2 for 2004 to 2006.

ies using model estimates in the Netherlands (Schils et al., 2005) and New Zealand (Basset-Mens et al., 2009) indicated that milk produced from WC has a CF between 10 and 15% lower than that from FN. Following from the study of Li et al. (2011), which was concerned solely with emissions of N₂O from WC and FN, the objective of the current study was to use LCA to conduct a holistic comparison of WC and FN using data from experimental systems at Solohead Research Farm to determine the difference in GHG emissions, including CO₂ and CH₄. Specific objectives were to (1) compare the CF of pasture-based milk production from FN and WC, (2) compare the modeled and measured N₂O emissions at the same site under similar management conditions, and (3) evaluate the sensitivity of the CF model.

MATERIALS AND METHODS

The 4 stages of LCA methodology: goal and scope definition, life cycle inventory, life cycle impact assessment, and result interpretation (ISO, 2006), were implemented as follows.

Goal and Scope

In the goal and scope phase, the production system is described, a functional unit (FU, to which all subsequent inputs and outputs are related) and system boundary (which determines the processes associated with the delivery of the FU) are defined, and in case of multiproduct systems, the allocation procedure between products is specified (ISO, 2006). The goal of the current study was to assess the role of CF for comparing milk production using FN and WC management in low-cost, grass-based, rotational grazing systems. Management data were obtained from experimental

systems at Teagasc Solohead Research Farm (52°51' N, 08°21' W) between 2001 and 2006 (Table 1; Humphreys et al., 2008, 2009). The first experiment had 4 systems (1 WC and 3 FN) in 2001 and was replicated in 2002. The second experiment had 2 systems (1 WC and 1 FN) in 2003 and was replicated in 2004, 2005, and 2006, resulting in 16 data sets being analyzed for this study. The LCA model was developed in Simapro (PRé Consultants, 2011) and was replicated for each system separately. The FU was defined as 1 kg of ECM at the farm gate (Sjaunja et al., 1990):

$$\begin{aligned} \text{kg of ECM} = & \text{kg of milk} \times (0.25 + 0.122 \\ & \times \text{Fat \%} + 0.077 \times \text{Protein \%}), \end{aligned} \quad [1]$$

where kg of milk is the total milk delivered from the herd in 1 yr. To account for on-farm consumption by calves, 301 kg of milk per cow was subtracted from the milk yield each year (O'Mara, 2006).

The system boundary was cradle-to-farm gate, including the foreground processes of milk production on the farm and the background processes of production and transportation of mineral fertilizer; cultivation, processing, and transportation of concentrate feed (except citrus pulp and minerals due to lack of data); production and use of electricity and diesel fuels; and clover seed. Infrastructure (sheds, slurry lagoon, roads), machinery (tractor, milk cooling system), medicines, pesticides, and disposal of plastic for baled silage were not included due to lack of relevance for comparison or because they were shared by both FN and WC management. Bulls were not accounted for because cows were inseminated by AI. Soil carbon sequestration was not included because of a lack of site-specific data, but the effect was considered during the interpretation phase.

Two parts of the milk CF calculation required allocation: between concentrate feed ingredients and co-products (pre-farm) and between milk and meat (on-farm). Pelletier and Tyedmers (2011) argue that economic allocation is not appropriate because market price does not reflect environmental concerns. However, the concentrate consumption was small (which reflected the practice of average Irish dairy farms) so economic allocation for concentrate feed was used, which is common practice (Yan et al., 2011). Data on sugar and oil co-products (rape meal, palm kernel meal, and molasses) were allocated by Ecoinvent version 2.2 (Ecoinvent, 2011), and soybean hulls and maize gluten meal were derived from Ecoinvent 2.2 data sets (soybeans, at farm; maize starch, at plant) and allocated according to Thomassen et al. (2008) and Cederberg and Mattsson (2000). Allocation between milk and meat (from surplus calves and culled cows) was carried out using an economic method based on market prices between 2000 and 2006 (<http://www.cso.ie/px/pxeirestat/Database/eirestat/AgriculturalInputandOutputAbsolutePrices/AgriculturalInputandOutputAbsolutePricesstatbank.asp?SP=AgriculturalInputandOutputAbsolutePrices&Planguage=0;http://epp.eurostat.ec.europa.eu/portal/page/portal/agriculture/data/database>). The effect of allocation between milk and meat was considered during the interpretation phase.

Life Cycle Inventory

In the life cycle inventory phase, all input and output processes are defined and quantified (ISO, 2006). The GHG inventory was made by multiplying life cycle activity data by emission factors (**EF**) derived from the literature (Table 2), and all emissions were calculated on a calendar year basis. Enteric CH_4 from cows was determined by estimating net energy (**NE**) for maintenance, lactation, and pregnancy (Shalloo et al., 2004; O'Mara, 2006). The NE of grazed pasture was calculated as the difference between NE intake from silage and concentrate and that needed to meet total NE requirements (Humphreys et al., 2008). The NE of feed was converted into DMI and multiplied by the EF for enteric fermentation of 21.6 g of CH_4 per kg of DMI (O'Mara, 2006). Replacement animals (calves and heifers) were not part of the systems (Humphreys et al., 2008, 2009) but would be necessary for the dairy unit to function. An average replacement rate of 23%, which was the average replacement rate on the farm during the period, was used to calculate the number of replacement units (1 calf and 1 heifer calving at 24 mo of age) in each system. Enteric CH_4 from replacement animals was estimated using national average EF (McGettigan et al., 2010). Emissions of CH_4 from manure manage-

ment were estimated by Intergovernmental Panel on Climate Change (**IPCC**) Tier 2 methods (McGettigan et al., 2010). Emissions of CH_4 from excreta deposition was estimated by multiplying EF for excreta deposition in the field with outdoor days. Emissions of CH_4 from slurry storage were added up from each storage period. Farmyard manure (**FYM**) consisted of manure and rejected silage (~93 kg/cow per year) and was land spread after about 180 d of storage. Emissions of CH_4 from FYM storage were estimated by multiplying EF by days stored and volume.

Direct N_2O emissions from manure management and soils were estimated using the methodology of McGettigan et al. (2010; Table 2). Excretion of N was estimated as the difference between intake of N and N deposited in milk, in calves, or in BW change of the cows (Humphreys et al., 2008). The N excretion of replacement animals were estimated as proportionate to dairy cow according to Statutory Instrument No. 610 (European Communities, 2010, p 38–39). About 36% of the N was excreted during the housing period (including milking time), 94% of which was assumed to be stored as slurry, whereas the rest went into FYM (McGettigan et al., 2010). After excluding NH_3 and NO_x volatilization according to McGettigan et al. (2010), the remainder of the N was multiplied by relevant EF to obtain the direct N_2O emissions on farm (Table 2).

Indirect N_2O emissions from NH_3 deposition and leaching were estimated using IPCC Tier 2 methods (McGettigan et al., 2010). Diesel combustion due to on-farm fieldwork was estimated from Kramer et al. (1999). Emissions of CH_4 and N_2O from the concrete yard were excluded because of their very small contributions (Casey and Holden, 2005). No appropriate EF were found for CH_4 or N_2O emission from dirty water storage and spreading, so they were not included.

Emission factors for background data (except electricity production) were taken from Ecoinvent v. 2.2 (Ecoinvent, 2011). Data sets for GHG emissions from production of calcium ammonium nitrate (**CAN**) and urea were selected because they were the 2 types of N fertilizers used on the research farm. A standard concentrate formulation was obtained from feed suppliers, and data sets regarding producing and processing relevant ingredients were selected from Ecoinvent v. 2.2 (Ecoinvent, 2011) to calculate the GHG emissions (including that from land use change). Electricity use in the systems was estimated from whole-farm records (in kWh, deducting that for office use and other equipment) and proportioned by the number of dairy cows because electricity is mainly used for milking and cooling of milk. Emissions associated with electricity use in each system were taken from Irish energy reports as the average of 2001 to 2006 (0.68 kg of $\text{CO}_2\text{-eq/}$

Table 2. On-farm emission factors used in the modeling for fertilizer N (FN)- and white clover (WC)-based systems

Item	CO ₂	CH ₄	N ₂ O
Enteric fermentation			
Dairy cows ¹		21.6 g/kg of DMI	
0–1 yr ²		27.79 kg/(head, yr)	
1–2 yr ²		45.39 kg/(head, yr)	
Manure management			
Slurry storage ^{2,3}		0.0082 kg/(m ³ , d)	0.001 kg of N ₂ O-N/(kg of N _{excreted} in slurry – kg of NH ₃ -N)
Farmyard manure (FYM) storage ^{2,3}		0.0059 kg/(m ³ , d)	0.02 kg of N/(kg of N _{excreted} in FYM – kg of NH ₃ -N + kg of N _{rejected} silage)
Emission from soil			
Excreta in field ^{2,4}		0.001683 kg/(cow, d)	0.02 kg of N ₂ O-N/(kg of N _{excreted} in field – kg of NH ₃ -N)
Slurry spreading ^{2,5}		0.00007 – 0.0123 kg/m ³	Slurry, FYM, and fertilizer: 0.0125 kg of N ₂ O-N/(kg of N – kg of NH ₃ -N and NO _x -N)
FYM spreading ^{2,5}		0.0027 kg/kg	
Fertilizer spreading ²			
Indirect N ₂ O emission ²			
From NH ₃ deposition			Volatilized NH ₃ × 0.01
From nitrate leaching			N input to soil × 0.1 × 0.025
Diesel combustion ⁶	3.56 kg/kg	0.00052 kg/kg	0.0007 kg/kg

¹O'Mara (2006).²McGettigan et al. (2010).³Husted (1994).⁴Jarvis et al. (1995).⁵Chadwick et al. (2000).⁶Kramer et al. (1999).

kWh; Howley et al., 2008), which was considered more reliable than the 0.216 kg of CO₂-eq/kWh from Eco-invent v 2.2 (Ecoinvent, 2011). All transportation was assumed to start from origin (fertilizer from Germany, clover seeds from Denmark and New Zealand, and concentrates from various places) and as necessary through Rotterdam and inland shipping to Dublin, Ireland, and then to the research farm.

Life Cycle Impact Assessment

In the life cycle impact assessment stage, the inventory is linked to specific environmental impact categories and indicators (ISO, 2006). To express the impact of GHG emissions, a commonly used indicator is global warming potential (**GWP**) defined for a period of 100 years. The mid-point method IPCC 2007 GWP 100a v 1.02 (developed by the IPCC; Forster et al., 2007) was selected in Simapro (PRé Consultants, 2011), which defined the GWP of CO₂ as 1, that of CH₄ as 25, and that of N₂O as 298. The total emissions of GHG were determined as follows:

$$\text{GHG} = \sum \text{GWP}_i \times m_i, \quad [2]$$

where m_i is the mass (kg) of the emitted gas i , where i refers to CO₂, CH₄, or N₂O. Finally, CF was calculated using the GHG allocated to milk production:

$$\text{CF} = \text{GHG allocated to milk production} / \text{FU}. \quad [3]$$

Interpretation

Interpretation of the life cycle impact assessment consists of identifying significant issues, evaluating sensitivity, and drawing conclusions (ISO, 2006). The CF of milk production from WC and FN throughout the 6 yr was compared using the *t*-test. Geographical origin of emissions (i.e., within or outside Ireland) and major components of CO₂, CH₄, and N₂O emissions were identified. Modeled N₂O emissions were compared with measurement at Solohead under similar stocking rate and fertilizer N. The relationship between surplus N (including clover BNF; Humphreys et al., 2008) and CF was studied. Because of the linear characteristic of the CF model, the main contributors (accounting for 95% of the GHG) had large influences and thus the relevant assumptions and parameters were evaluated by assessing their tenability with ratio sensitivity analysis (Björklund, 2002). Ratios were calculated by determining the percentage that those assumptions and the EF needed to change to reverse the ranking of WC and FN with respect to CF. The uncertainty ranges of EF from the literature were used to gain insight into the tenability of the ratio change. The ratio sensitivity analysis was mainly conducted with EF rather than activity data because most activities of WC and FN management were the same. Thus, the aim of the ratio sensitivity was primarily to look at the effect of the uncertainty of EF on the comparison between the systems. Comparisons with similar Dutch and New

Table 3. Process contribution and ratio sensitivity analysis (kg of CO₂-eq/kg of ECM; % contribution) for the fertilizer N (FN)- and white clover (WC)-based systems

Item ¹	System		Relevant assumptions and emission factors (EF)	Ratio sensitivity analysis, ² %	Range of EF, ³ %
	WC	FN			
Enteric fermentation	0.46 (53)	0.46 (45)	EF of CH ₄ Replacement rate	+23 to 55 +100 to 190	–/0 NA ⁴
Manure management					
Slurry storage	0.03 (3)	0.03 (3)	EF of CH ₄ EF of N ₂ O	+368 to 897 +6,009 to 14,483	+320 +100
Farmyard manure (FYM) storage	0.01 (1)	0.01 (1)			
Emission from soil					
Excreta deposition	0.12 (14)	0.12 (12)	EF of CH ₄ EF of N ₂ O	+6,116 to 14,920 +98 to 239	–17 +200
Slurry spreading	0.02 (3)	0.02 (2)	EF of CH ₄ EF of N ₂ O	+17,400 to 41,938 +148 to 375	NA +380
FYM spreading	0.00 (–)	0.00 (–)			
Fertilizer spreading	0.04 (5)	0.11 (10)	EF of N ₂ O	+148 to 375	+380
Indirect N₂O	0.05 (6)	0.06 (6)	EF of NH ₃ -N EF of NO ₃ -N	+565 to 1,367 +398 to 965	+400 +230
Diesel combustion	0.02 (2)	0.02 (2)			
Soil carbon sequestration	NA	NA	Omitted	WC loses 0.8 to 2.4 t of CO ₂ -eq/ha	NA
On-farm subtotal	0.75 (86)	0.83 (80)			
Fertilizer production	0.02 (3)	0.11 (10)	GHG per kg of fertilizer N	+413 to 998	Urea +8.4, CAN ⁵ +17.1
Concentrate production	0.03 (4)	0.03 (3)	GHG per kg of concentrate ingredient	+321 to 776	+6.2
Diesel production	0.00 (–)	0.00 (–)			
Electricity production⁶	0.05 (6)	0.05 (5)	GHG per kWh of electricity	+165 to 400	NA
Transportation	0.00 (1)	0.01 (1)			
Pre-farm subtotal	0.12 (14)	0.20 (20)			
Total (economic allocation)	0.87	1.04	Fixed at 91%	+12 to 30	

¹Components that accounted for 95% of total greenhouse gases (GHG) are shown in bold.

²Ratios were calculated by determining the percentage that those assumptions and EF needed to change to reverse the ranking of WC and FN with respect to carbon footprint.

³Range of EF suggested by the literature.

⁴NA = not available.

⁵Calcium ammonium nitrate.

⁶Only refers to the electricity used on farm; that used for production and transportation of fertilizer, concentrate, and diesel was embedded in relevant data sets of Ecoinvent (2011).

Zealand studies were made in terms of methodology and dairy systems.

RESULTS

Carbon Footprint of WC and FN

The CF of milk was found to be 11 to 23% lower on WC compared with FN across the range of fertilizer N inputs. With economic allocation (91% to milk), the average milk CF from WC was 0.86 to 0.87 kg of CO₂-eq/kg of ECM and that from FN was 0.97 to 1.13 kg of CO₂-eq/kg of ECM ($P < 0.001$; Table 3). A greater proportion of GHG emissions was from within Ireland and was higher from WC (85%) than from FN (80%).

On average, CH₄ emissions contributed 0.49 and 0.50 kg of CO₂-eq/kg of ECM to WC and FN ($P > 0.05$), the majority of which was from enteric fermentation, followed by manure management, and emissions from soil. Emissions of N₂O were significantly different between WC and FN (0.26 and 0.39 kg of CO₂-eq/kg of ECM, respectively; $P < 0.001$), the majority of which was from soils and production of concentrate. Because of the larger amount of fertilizer N applied in FN, a substantially larger share of pre-farm N₂O emissions was found for FN (19%) than for WC (8%); the on-farm N₂O emissions from WC and FN were 9.4 and 14.0 kg of N/ha per year, respectively. Likewise, CO₂ emissions were significantly different between WC and FN (0.12 and 0.14 kg of CO₂-eq/kg of ECM; $P < 0.001$), most

of which was from pre-farm production of fertilizer, concentrate, and electricity, and on-farm diesel combustion. Carbon dioxide from land transformation was mainly from palm fruit production for concentrate feed. Significant correlation was found between surplus N per kg of ECM and CF (Figure 1; $R^2 = 0.66$, $P < 0.001$), which indicated that a 1-g reduction of on-farm surplus N could reduce CF by 23 g of $\text{CO}_2\text{-eq.}$ From 2003 to 2006, when WC and FN had the same stocking density (2.0 and 2.2 cows/ha), the CF of WC was significantly lower than that of FN (Table 4).

Ratio Sensitivity Analysis of Assumptions and Parameters

The contributors that accumulated 95% of GHG (shown in bold in Table 3) were enteric CH_4 , excreta deposition, fertilizer spreading, fertilizer production, electricity production, indirect N_2O , slurry storage, concentrate production, and slurry spreading.

The largest contributing factor was enteric CH_4 , which was determined by EF of enteric CH_4 for each type of animal. To reverse the ranking of WC to FN (while keeping FN unchanged), the EF of enteric CH_4 in WC needed to be increased by 23 to 55% and the

Table 4. Carbon footprint (economic allocation between milk and meat) of fertilizer N (FN)- and white clover (WC)-based systems under relevant stocking rates

Stocking rate, livestock unit/ha	Carbon footprint, kg of $\text{CO}_2\text{/kg}$ of ECM	
	WC	FN
1.75	0.89	
2.1		1.00
2.0	0.88 ^a	1.05 ^b
2.2	0.86 ^a	1.01 ^b
2.5		1.07 ¹
2.5		1.16 ²

^{a,b}Means within a row with different superscripts differ ($P < 0.001$).

¹Fertilizer N rate of 248 kg of N/ha.

²Fertilizer N rate of 353 kg of N/ha.

replacement rate by 100 to 190% (depending on N input in FN).

The second largest contributor to CF was N_2O from N deposition on pasture. The third largest contribution was from spreading fertilizer N, which consisted of N_2O from urea and CAN. To reverse the ranking of WC and FN, the EF for N_2O emissions from N deposition on pasture in WC needed to be increased by 98 to 239% (while keeping the N excretion unchanged), the EF of CH_4 from excretion deposition needed to be increased by 6,116 to 14,920%, or the EF for fertilizer spreading needed to be increased by 148 to 375%.

Soil carbon sequestration was not considered and the same allocation factors were used. To reverse the ranking of WC and FN, WC needed to lose 0.8 to 2.4 t of $\text{CO}_2\text{-eq/ha}$, or the economic allocation factor needed to be increased by 12 to 30%. All other contributors were small, and thus large changes were required for the ratio sensitivity analysis (Table 3) and they were much larger than the uncertainty ranges in the literature.

DISCUSSION

Comparison with Similar Studies

A range of annual weather conditions (rainfall from 796 to 1,149 mm) and management practices (stocking density from 1.75 to 2.5 cows/ha; fertilizer application rate from 80 to 353 kg of N/ha) were encompassed by the data sets used in this study. The difference in milk CF between the systems was in general agreement with other similar studies. For example, Schils et al. (2005) found that GHG emissions per kilogram of milk from a grass-clover system were 10% lower than those from a grass-fertilizer N system. The methodology was similar to that used in the current study although it was referred to as whole-farm accounting. The main differences in the Schils et al. (2005) study were that

Figure 1. Relationship between on-farm surplus N (kg of N/kg of ECM, $\times 0.01$) and carbon footprint (kg of $\text{CO}_2\text{-eq/kg}$ of ECM, economic allocation, $r = 0.6$, $P < 0.001$) of fertilizer N (FN; Δ)- and white clover (WC; $\circ$)-based systems.

carbon sequestration was 24% lower in the grass-clover system because of the greater proportion of plowing, and BNF was included as a source of N_2O emission (5 g of N/kg of N fixed), both of which reduced the difference in milk CF between WC and FN. Recent research has shown that direct N_2O emission from BNF per se is thought to be negligible and the N_2O emissions induced by the growth of legume crops and forages may be estimated solely as a function of the above-ground and below-ground N inputs from crop residues (Rochette and Janzen, 2005). Basset-Mens et al. (2009) followed the LCA approach and found that CF from a grass-clover system with zero fertilizer N was 15% lower than that from a grass-fertilizer N system under 139 kg of N/ha per year. Differences existed in their calculation of GHG inventory (e.g., EF for N_2O emission from excreta deposition was 1% instead of the 2% in this study) but should not affect the general conclusion.

In terms of dairy systems, there were large differences between Schils et al. (2005) and Basset-Mens et al. (2009). In Schils et al. (2005), stocking density was 1.9 to 2.2 livestock units/ha of high-yielding Holstein-Friesian cows (about 8,194 kg of milk/cow per year), with a short grazing season (April to September) and high concentrate feeding (approximately 1,823 kg/cow per year). In Basset-Mens et al. (2009), stocking density was high, 2.7 to 5.2 cows/ha of low-yielding Holstein-Friesian cows (approximately 4,845 kg of milk/cow per year), with a long grazing season (almost all year) and zero concentrate feeding. The system in the current study was intermediate in terms of milk output per cow, stocking rate, grazing period, and concentrate fed. Although direct comparison between LCA studies has been difficult (Yan et al., 2011), the 3 studies collectively show that WC reduces the CF of milk production under research conditions.

N₂O Emissions from WC and FN

The fundamental difference between the CF of milk production from WC and FN was the contribution of BNF. During 2004, 2005, and 2006, the WC and FN systems had the same stocking density (2.2 livestock units/ha) and achieved similar milk yields per cow (6,515 and 6,518 kg/cow per year), and the total N input from fertilizer, estimated BNF, and animal excretion to soils were 477 and 528 kg of N/ha, respectively (without excluding NH_3 -N loss during fertilizer spreading and manure storage). The modeled N_2O emissions were 10.7 and 14.2 kg of N/ha per year from WC and FN swards during these years. In other words, in the present study, WC had 9% lower total N inputs but was estimated to have 25% lower N_2O emissions from soils

than FN; the reason was that BNF was not considered to contribute to N_2O emissions (Rochette and Janzen, 2005). In a study by Li et al. (2011) involving the direct measurement of N_2O emissions from white clover and fertilizer N-based systems of dairy production (similar to those in the present study), annual emissions were 6.4 kg of N_2O -N/ha from the WC system and 7.8 kg of N_2O -N/ha from the FN system. A clear reduction in N_2O emissions was found when fertilizer N was replaced by BNF (Li et al., 2011), which was reflected in the difference in milk CF between the systems in this study.

Biological N fixation has been found to respond to changes in soil N, which could mean greater efficiency of N utilization in WC-based pasture (Ledgard and Steele, 1992). However, N use efficiency (NUE) may differ little at the farm gate (i.e., N output/N input). In grazing systems, NUE is closely related to optimization of the system inputs to the milk output (Fitzgerald et al., 2005) because urine during grazing (Rochette et al., 2004) and ammonia volatilization (Andrews et al., 2007) are critical to the N cycle and N_2O emissions from the system. Under similar levels of total N input (including BNF), WC may have the same NUE as FN (Schils et al., 2000), whereas if FN has higher N input, it tends to have lower NUE than WC (Ledgard et al., 1999). This is in agreement with results of the current study and those of Humphreys et al. (2008, 2009), in which WC had lower total N input, higher NUE, and lower CF than FN during 2001 to 2002, and had similar total N input and NUE but 15% lower CF than FN during 2003 to 2006.

Sensitivity at the Experimental System Scale

The ratio sensitivity analysis revealed that to reverse the ranking of WC and FN, changes to EF and assumptions had to be much greater than the uncertainty ranges found in the literature (Table 3). For example, the EF of enteric CH_4 in WC would have to be increased by 23 to 55%, but enteric CH_4 from cows fed white clover was found to be lower (Pinares-Patiño et al., 2009) or similar (van Dorland et al., 2007) to that of cows fed grass. Because of the similar digestibility of the grazed pasture during the studies (Humphreys et al., 2008, 2009), the same EF of enteric CH_4 was applied to both WC and FN. During earlier years, however, the pre-grazing herbage of WC tended to have lower (although not always the lowest) OM digestibility (Humphreys et al., 2008). The required 23 to 55% increase would mean either higher gross energy content of clover or a larger methane conversion factor (IPCC, 2000), neither of which is likely (Andrews et al., 2007). On the other hand, the EF of CH_4 from excretion deposition in WC

would have to be increased by 6,116 to 14,920%, but Jarvis et al. (1995) suggested a lower CH₄ emission rate from dairy cows grazing grass or clover swards.

Other examples from the ratio sensitivity analysis support the conclusion that the difference between the management approaches was not caused by error. For instance, increasing the economic allocation in WC by 12 to 30% would result in no economic value for the slaughtered animals from WC, which is not tenable. Moreover, in Europe, permanent grasslands are believed to be carbon sinks, where higher N input will increase carbon sequestration (Soussana et al., 2007); therefore, it is not tenable that WC could lose 0.8 to 2.4 t of CO₂-eq/ha from oversowing of clover swards, which does not involve cultivation. There is no reason to believe that a significant difference in abiotic carbon storage between the 2 sward management approaches would arise in the timescale of these experiments. However, at the lower fertilization rate in FN (during 2001 and 2002), the difference in milk CF between WC and FN was less certain because of the large variation of EF for N₂O emissions from soils, which was thought to follow a log normal distribution from 0.25 to 6%, with a default of 1.25% (IPCC, 2000).

CONCLUSIONS

Life cycle assessment of pasture-based milk production from WC and FN systems indicated that the CF for WC was 11 to 23% lower (per kg of ECM) than that for FN, which suggested that clover could be used to reduce the CF of milk. Ratio sensitivity analysis showed that enteric fermentation, N₂O emissions from soils, soil carbon sequestration, and allocation could influence, but not reverse, the ranking of WC and FN systems, and that introducing clover reduced CF at the experimental system scale.

ACKNOWLEDGMENTS

This work was supported by the Department of Agriculture and Food Research Stimulus Fund Programme (RSF07-516) funded by the Irish Government National Development Plan, and funded in part by the European Research and Development Fund (ERDF) via Interreg IVB project O96D: Dairyman.

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